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Coastal wetland rehabilitation first-pass prioritisation for blue carbon and associated co-benefits

Kerrylee Rogers[A,*], Kirti K. Lal[A,B], Emma F. Asbridge[A] and Patrick G. Dwyer[C]

For full list of author affiliations and declarations see end of paper

*Correspondence to:
Kerrylee Rogers
School of Earth, Atmospheric and Life Sciences, University of Wollongong, Wollongong, NSW 2522, Australia
Email: kerrylee@uow.edu.au

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ABSTRACT

Context. The Australian Government has developed a methodology for payment for carbon services provided by blue carbon ecosystems that focuses on avoided emissions and carbon additionality resulting from tidal restoration of coastal wetlands. **Aims.** This study is a first-pass prioritisation for tidal restoration of coastal wetlands in New South Wales (NSW). **Methods.** A pixel-based approach was applied using readily available datasets, with particular focus on watersheds above in-stream tidal barriers. **Key results.** Many sites were identified, to investigate in detail, opportunities to restore tidal flows to coastal wetlands. More were associated with the broad coastal floodplains of northern NSW than narrower floodplains of southern NSW. **Conclusions.** Information is needed about the location, ownership, land tenure, structure, condition and height of in-stream and over-land flow barriers, particularly in the context of rising sea levels. Decisions about managing in-stream drainage and flood mitigation infrastructure should be made cognisant of opportunities to increase blue carbon, and provide associated co-benefits, including mitigating other deleterious impacts from coastal wetland drainage. **Implications.** Decision support tools for evaluating economic and environmental costs and benefits of tidal barriers will assist decision-makers assessing future proposals to repair or remove aging barriers, or create new tidal barriers.

Keywords: acid sulfate soils, blue carbon markets, coastal floodplains, coastal wetlands, mangroves, saltmarshes, tidal barriers, tidal reintroduction.

Introduction

Low-energy intertidal environments support coastal wetlands dominated by mangroves and saltmarshes and provide many ecosystem services, including coastal protection, wildlife habitat, nutrient cycling and carbon storage (Barbier *et al.* 2011; Costanza *et al.* 2014). Carbon storage, sequestration and cycling services provided by coastal wetlands, known collectively as blue carbon, are receiving considerable scientific interest because carbon stocks may be an order of magnitude higher than tropical rainforests and other terrestrial ecosystems (Murray *et al.* 2011). In addition, blue carbon provided by coastal wetlands has received government interest because of the urgent need to mitigate atmospheric carbon (Kelleway *et al.* 2017, 2020). For this to be achieved using blue carbon, and presuming that methane emissions are limited, carbon addition to coastal wetlands must exceed carbon losses to attain a net increase in carbon stocks.

Globally, there has been significant deceleration in the loss of coastal wetlands by conversion to other land-uses, such as shrimp aquaculture, coastal developments, forestry, and palm oil plantations (Friess *et al.* 2019, 2020). This may be attributed to restoration, improved conservation and technological advancements in quantifying wetland extent (Finlayson and Gardner 2021), with the Ramsar Convention on Wetlands playing a critical role in improving the profile of wetlands. In Australia, policy has been implemented at national and state levels, to halt the decline in coastal wetland extent (Rogers *et al.* 2016*a*). Such efforts have arrested carbon emissions from the conversion of coastal wetlands and maintained carbon services. However, they do little to foster carbon drawdown and climate mitigation efforts. Some carbon additionality occurs as coastal wetland vegetation grows and adds biomass and through the accumulation of mineral and organic material within substrates. However, this additionality is reasonably minor and may be offset by natural processes of organic matter decomposition. Effectively harnessing the carbon services provided by coastal wetlands to achieve carbon additionality requires an increase in the three-dimensional space occupied by coastal wetlands (Rogers *et al*. 2019*a*). Because they occur at the interface between the land and the sea, an increase in lateral extent can be facilitated by increasing the area of tidal inundation, or by facilitating the vertical growth of substrates and sequestration of organic material. Fortunately, there remains considerable capacity for additionality to be achieved (Rogers *et al*. 2019*b*), and policy is being established to facilitate this process. The United Nations Framework Convention on Climate Change (UNFCCC) includes mechanisms relevant to blue carbon ecosystems including Reducing Emissions from Deforestation and Forest Degradation (REDD+) and the Clean Development Mechanism (CDM). In addition, Resolution XIII.14 of the Ramsar Convention attests to the role of the Ramsar Convention in meeting UNFCCC objectives, and explicitly promotes conservation, restoration and sustainable management of mangrove and saltmarsh blue carbon ecosystems. It promotes prioritisation of blue carbon ecosystems, and development and implementation of plans for conservation, restoration and sustainable management. This is achieved by encouraging contracting parties with blue carbon and associated ecosystems to maintain their ecological character, pursue policies and projects to conserve and restore blue carbon ecosystems, raise awareness of blue carbon ecosystems, and collect and analyse data about blue carbon ecosystems. It also encourages contracting parties to manage ecosystems consistent with the *Principles and guidelines for incorporating wetland issues into Integrated Coastal Zone Management* (i.e. Resolution VIII.4), promote dialogues among stakeholders, facilitate sharing of information, and develop and implement plans for conservation, restoration and sustainable management, and maintain and restore blue carbon ecosystems affected by coastal infrastructure.

Coastal wetlands occupy low-lying, often highly fertile land, and many have been converted to other land-use or affected by encroaching urbanisation, industrial developments, and agriculture (Rogers *et al*. 2016*a*). This is particularly the case in eastern Australia where large-scale programs to drain coastal wetlands and facilitate conversion to other land-uses occurred between the 1900s and 1980s (Goodrick 1970; Saintilan and Williams 2000; Sinclair and Boon 2012; Creighton *et al*. 2015), ceasing only when effective legislation prohibiting loss of coastal wetlands was enacted (Rogers *et al*. 2016*a*). These programs of wetland drainage, often under the guise of flood mitigation works, resulted in engineered works and structures being established to facilitate drainage (e.g. ditches, ring drains, floodgates, redirection, straightening and deepening of floodplain waterways) and impede tidal exchange (e.g. barrages, culverts, bunds, dykes, levees; Tulau 2011). In New South Wales (NSW), 4200 structures are estimated to impede flows in coastal rivers and streams (Williams and Watford 1997). The drainage of freshwater wetlands on the floodplain and exclusion of tidal exchange has changed wetland hydroperiod. This has facilitated the conversion of saline wetlands on land behind these structures into freshwater wetlands or pasture suitable for grazing and cropping. Inundation regimes for freshwater wetlands have reduced from >100 days to generally <10 days, enabling establishment of introduced pasture grasses to facilitate their conversion to agricultural landscapes (Tulau 2011). Rogers *et al*. (2016*a*) calculated the loss of potential fish habitat (including mangrove and saltmarsh) by drainage in the north-coast region assessed by Goodrick (1970), finding that 62 258 ha were drained since European settlement, constituting over 70% of the pre-European extent of 87 008 ha.

Implementation of wetland drainage and flood mitigation works has generated environmental disservice. Some of the land that is now cut off from tidal exchange may have converted from carbon sinks into likely sources of methane emissions (Poffenbarger *et al*. 2011). This is particularly concerning, given the 25–100 times greater radiative forcing of atmospheric methane than carbon dioxide (Kroeger *et al*. 2017). Estuary-wide water-quality impacts can occur with the exposure of potential acid sulfate soils to aerobic conditions and the consequential activation of acid sulfate soils and generation of acid discharges (Sammut *et al*. 1995, 1996). Likewise, there are episodic releases of deoxygenated blackwater from drained wetlands into estuaries when flood-intolerant pasture grasses, weeds and up-slope native vegetation that have established in the drained dryer wetland are inundated, drown and rot during large flood events (Wong *et al*. 2011). Waterways artificially disconnected from tidal flows may also be sites where monosulfidic black ooze is generated (Bush *et al*. 2004) or where eutrophication may begin to dominate (Lovelock *et al*. 2009). From an agricultural production perspective, the resulting rapid loss of organic material within substrates, that will likely have increased methane and carbon dioxide emissions, has also contributed to loss of substrate volume and elevation (Belperio 1993), with the outcome being that once profitable agricultural land becomes increasingly less viable for grazing and agricultural purposes. In some situations, this has increased exposure of peat substrates to fire, which can cause significant additional slumping of ground surfaces. In many places, the benefits of wetland drainage works are no longer being realised, and efforts are now being directed to rehabilitate coastal wetlands to re-establish environmental services. This involves reinstating former groundwater levels, sometimes by filling in constructed drains, and returning tidal exchange in natural waterways. For example, rehabilitation activities have occurred in the formally drained Yarrahapinni Wetland on the Macleay, where a floodgate and in-stream bund were removed, restoring tidal flows into over 700 ha of drained estuarine wetland, whereas over 200 ha of drained freshwater and brackish wetland at Big Swamp on the Manning River has undergone drain filling and tidal reinstatement (Rogers *et al.* 2016*b*). In some cases, restoration of tidal exchange has been facilitated by the failure of engineered drainage structures (P. G. Dwyer, pers. comm.); it is evidently difficult to hold back the sea indefinitely. Elsewhere, the benefit achieved from drainage works is diminishing because sea-level rise increases the elevation of tidal planes, and existing engineered structures may not effectively drain or impede tidal exchange in the future (Hanslow *et al.* 2018; Hague *et al.* 2020).

Aging engineered structures, many of which are deteriorating or no longer meet design expectations of draining wetlands, may provide an opportunity for blue carbon additionality. Areas previously cut off from tidal exchange will offer the much-needed space for blue carbon ecosystems to re-establish and increase carbon sequestration and storage. Indeed, for some very low-lying coastal floodplains, blue carbon restoration opportunities may become the most viable land-use option as sea-level rise continues further, compromising drainage infrastructure. In addition, restoration may reverse the environmental disserves associated with drainage, and provide environmental, economic, social, and cultural benefits beyond carbon abatement; these additional services associated with blue carbon restoration are commonly identified as 'co-benefits' within carbon abatement schemes. In Australia, the Commonwealth Government has pursued a blue carbon mechanism that contributes to Australia's climate mitigation efforts (Australian Government Clean Energy Regulator 2016). Administered by the Commonwealth Government Clean Energy Regulator, the Emissions Reduction Fund (ERF) provides a payment for carbon additionality that is adequately verified. A methodology has now been developed and implemented for quantifying blue carbon resulting from activities that promote carbon additionality by removing barriers to tidal exchange (Clean Energy Regulator 2022). This methodology stipulates the accepted approach for quantifying carbon abatement from restoration activities and provisions the supply of Australian Carbon Credit Units (ACCUs) to a person undertaking the restoration activity. An ACCU represents 1 tonne (Mg) of carbon dioxide equivalent (CO_2-e) stored and avoided by a tidal restoration project.

Prioritising areas suitable for blue carbon tidal restoration remains a critical knowledge gap that is receiving increasing attention (Moritsch *et al.* 2021; Duarte *et al.* 2022). Rogers *et al.* (2019*b*) developed a spatial framework for assessing blue carbon stocks and additionality that relied on accessible spatial datasets that were analysed using an indicator-based approach. Recognising geomorphological control on the distribution of blue carbon ecosystems and the preservation of sequestered carbon, the broad-scale approach included a first-pass assessment of the capacity for blue carbon storage, preservation, generation and permanency within coastal landscapes. This prioritisation was moderated on the basis of whether current land-use activities were compatible with the blue carbon services. However, it did not explicitly consider the role of wetland drainage and flood mitigation activities in moderating blue carbon services. In this study, we apply the blue carbon spatial framework with the intent of identifying the floodplain areas affected by wetland drainage and flood mitigation works because they could be used to prioritise restoration of tidal flows for blue carbon opportunities. Unlike Rogers *et al.* (2019*b*), this study includes information from the NSW Government fish passage-barrier database to identify locations on the northern and southern coasts of NSW where interventions could be undertaken to achieve blue carbon outcomes, thereby aligning with the methodology supported by the ERF. We anticipate that application of this framework to prioritise areas for restoration of tidal flows for blue carbon will provide additional confidence when considering sites and activities to meet blue carbon objectives of climate mitigation and coastal wetland rehabilitation generally. There are many considerations involved in coastal wetland rehabilitation and further analyses of costs and benefits associated with restoration would be required prior to selecting sites for restoration activities. This framework is applied to the northern and southern coasts of NSW but excludes the metropolitan region of Sydney (south of Tuggerah Lake to north of Lake Illawarra) where the subsurface mapping of the coastal Quaternary geology is less reliable owing to the lack of field validation to resolve uncertainty in areas where anthropogenic reworking of surface veneer sediments occurred (Troedson and Deyssing 2015). In doing so, the analysis excludes the tide-dominated drowned river valley estuaries that dominate the Sydney metropolitan area, although the drowned river valley of Batemans Bay and the large embayment of Jervis Bay and Twofold Bay remain within the analysis. The spatial framework is defined on the basis of geomorphological control of blue carbon, and applying the framework to the northern and southern coasts of NSW provides the opportunity to consider the implications of wetland drainage and flood mitigation activities on the predominantly wave-dominated estuaries that occur along these coasts.

Study location: wave-dominated coastline of NSW

This study focuses on the predominantly wave-dominated estuaries that occur on the NSW northern coast, extending from the catchment of the Tuggerah Lake to the northern

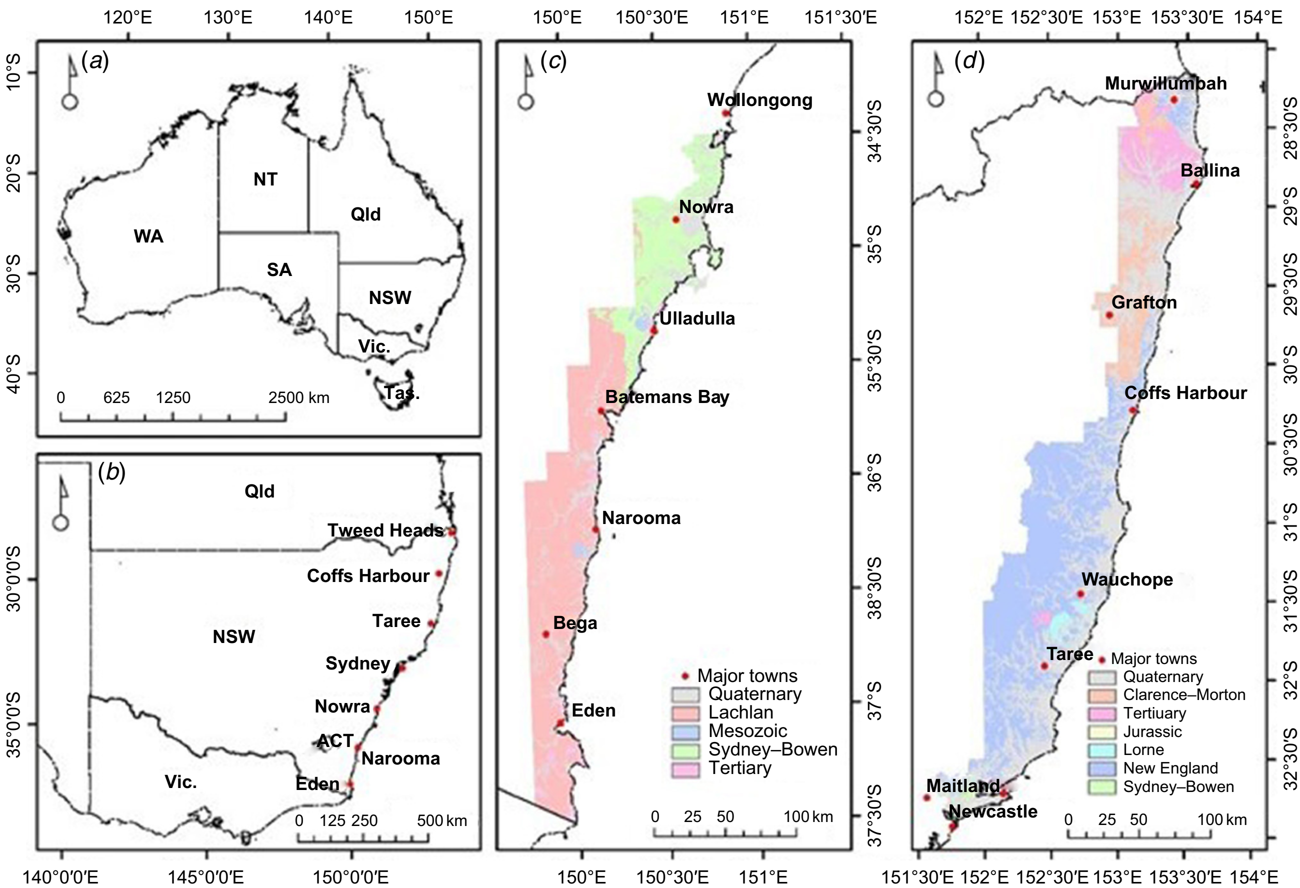

Fig. 1. Location map of (*a*, *b*) New South Wales, with bedrock, tertiary and quaternary geology of the (*c*) south-eastern and (*d*) north-eastern coast, indicating the spatial extent of available datasets. Source: Troedson *et al.* (2004).

border of NSW, and the NSW southern coast, extending from the catchment of Lake Illawarra to the southern border of NSW (Fig. 1, Supplementary Fig. S1). As sea level increased since the last glacial maximum, coastal embayments were drowned and coastal barriers that formed along the coast restricted tidal exchange between fluvial and oceanic water, resulting in the formation of wave-dominated estuaries.

Influence of coastal geomorphology on blue carbon in wave-dominated estuaries of NSW

Coastal blue carbon ecosystems occur within the intertidal zone of low-energy shorelines, and are usually positioned above mean sea level. Along the wave-dominated coastline of NSW, Australia, suitable conditions are typically restricted to estuaries where entrances provide shelter from the high-energy waves of the open coast. Roy *et al.* (2001) classified estuary structure for south-eastern Australia on the basis of (1) being wave- or tide-dominated, and (2) the degree of sediment infill that has occurred since their formation, known as estuary maturity. This geological classification recognises that the coastal zone was located on the continental shelf during glacial periods, and coastal valleys drowned during interglacial periods (Fig. 2*a*). Near the end of the last marine transgression when sea-level rise decelerated and stabilised (i.e. *c.* 7000 years ago; Lewis *et al.* 2013), coastal barriers along the high-energy coastline enclosed many of the shallow drowned river valleys, creating estuaries (Fig. 2*b*) (Roy 1984; Sloss *et al.* 2005, 2010), and most of the estuaries of south-eastern Australia are classified as wave-dominated. Only the deepest drowned river valleys, located in the Sydney metropolitan area and Batemans Bay, are regarded to be tide-dominated (Roy *et al.* 2001). Since the early Holocene, estuaries have been infilling with both terrigenous sediments and marine sediments. Variation in the rate of sediment supply, and the size of estuaries means that estuaries can range in the degree of infill from immature stages consisting of a large waterbody (e.g. lake) and narrow coastal and alluvial floodplains, to mature estuaries that have channels traversing broad coastal floodplains (Roy *et al.* 2001). Estuary structure (type and maturity), waterbody size and catchment area have a profound influence on coastal blue carbon.

In the early stages of wave-dominated estuary evolution (i.e. immature estuaries), streams deliver terrigenous sediment from catchments to the open waters of estuaries

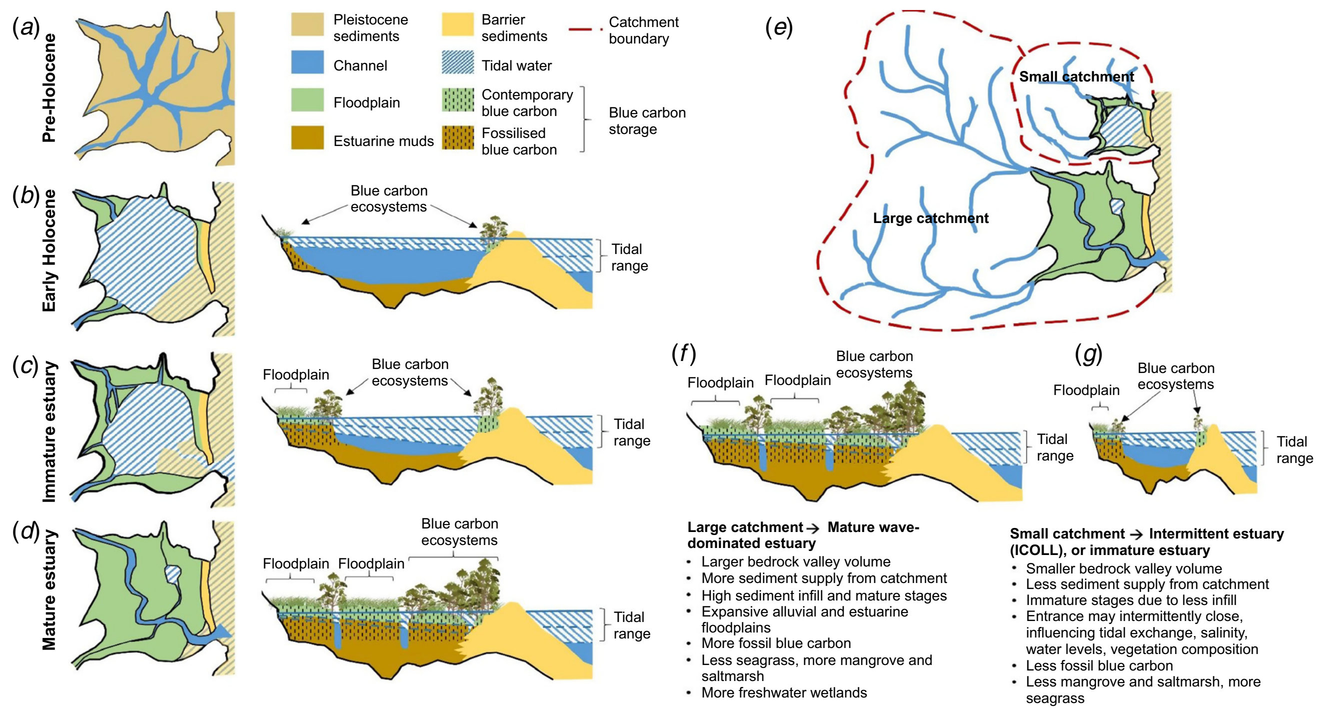

Fig. 2. Conceptual models of the evolution of wave-dominated estuaries from the (*a*) pre-Holocene, (*b*) early Holocene, and eventually to (*c*) immature and (*d*) mature stages of infilling with sediments; and the influence of estuary maturity on blue carbon ecosystem extent, and carbon storage; and conceptual models of the influence of (e) catchment size on estuary function, blue carbon ecosystem extent and carbon storage, particularly with reference to (*f*) large and (*g*) small catchments. Adapted from Roy *et al.* (2001).

and tides deliver marine sediment through estuary entrances (Fig. 2*c*). Because hydrological energy diminishes when streams enter open waters, sediment falls from entrainment and fluvial deltas form. Similarly, entrained marine sediments delivered through estuary entrances on tides also accumulate where hydrodynamic energy diminishes, and contribute to the development of a flood-tide delta (Roy 1984; Roy *et al.* 2001). Three broad depositional environments may establish, including coastal barrier, estuarine plain and alluvial plain. Fluvial and flood-tide deltas, and back barrier substrates provide favourable intertidal conditions for coastal wetland vegetation to establish and thrive. The intertidal zone within immature estuaries and the vertical distribution of coastal wetland vegetation are controlled by the influence of estuary entrance morphology on the tidal prism; constriction of the prism typically results in tidal range being diminished as tides propagate into open waters.

As an estuary infills with sediments, fluvial deltas and flood-tide deltas encroach upon open estuarine waters; the area of open water diminishes, and floodplains develop and broaden (Fig. 2*c*). The broadening of coastal floodplains and greater areal extent of the intertidal zone support more expansive intertidal coastal wetlands (Roy *et al.* 2001). The ensuing accumulation of organic material within sediments baffles hydrodynamic energy, enhances sedimentation, binds sediments and buffers erosion, creating a feedback that promotes accumulation of organic-rich material within substrates (Rogers *et al.* 2017). Over time, intertidal substrates increase elevation, and older organic material (roots) are increasingly buried (McKee 2011; Woodroffe *et al.* 2016). Termed 'fossil' blue carbon (Rogers *et al.* 2019*b*), this preserved carbon will undergo decomposition at rates that are time-dependent and influenced by substrate salinity and oxygen availability. More specifically, decomposition of fossil blue carbon diminishes under anaerobic conditions created by tidal inundation and high groundwater levels, whereas methanogenesis is inhibited in saline substrates that arise from periodic saline tidal inundation (Duarte *et al.* 2005, 2013; Mcleod *et al.* 2011; Macreadie *et al.* 2017*b*). Decomposition is also related to variation in sediment characteristics across an estuary; finer grained silts and muds typical of fluvial deltas enhance anaerobic conditions that slow decomposition; sand-dominated sediments typical of flood-tide deltas and back barrier zones may have greater aerobic decomposition owing to more pore spaces (Saintilan *et al.* 2013; Kelleway *et al.* 2016).

In the final stages of maturity, open waters are restricted in extent and channels traverse floodplains comprising sediments that have infilled coastal valleys since the early Holocene (Fig. 2*d*; Roy 1984; Roy *et al.* 2001). Tidal wetlands will be more restricted and freshwater wetlands will occur where groundwater is at or near the surface, and fossil blue carbon that has accumulated within substrates over the Holocene may have had considerable time to undergo diagenesis (Rogers *et al.* 2019*a*). As tides deliver sulfates to substrates over millennia, 'fossil' blue carbon stores may convert to acid sulfate soils when exposed to aerobic conditions (Rosicky *et al.* 2004; Johnston *et al.* 2016). Preservation of saline anaerobic conditions serves to both preserve fossil blue carbon and prevent development of acid sulfate soils. Estuaries in mature stages tend to have the most extensive distribution of intertidal coastal wetland vegetation and broad coastal floodplains with freshwater wetlands (Rogers *et al.* 2019*b*).

Considerable variation in estuary size, waterbody area and catchment area occurs along the coastline of south-eastern Australia. Intermittently closed and open lakes and lagoons, commonly referred to as ICOLLS (Haines 2006; Haines *et al.* 2006; Maher *et al.* 2011), form 70 of the 135 estuaries of NSW. They occur in catchments that are relatively small in comparison to the estuary waterbody area and may be exposed to above-average wave energy at the coast. The combination of lower catchment in flows and higher wave energy facilitate episodic closure of estuary entrances. The distribution of coastal wetland vegetation and blue carbon services has been correlated with catchment area, whereby conditions favourable for blue carbon generation are positively correlated with catchment area (Rogers *et al.* 2019*b*). Catchment area also influences sediment availability and supply to estuaries, with infill over the Holocene being typically greater when catchments are large; accordingly, wave-dominated estuaries in the largest catchments have been classified as mature (Fig. 2*e*, *f*).

Materials and methods

Approach

This approach uses accessible spatial data sets that are reclassified and adapted to create raster datasets that indicate the present-day capacity for carbon storage, preservation, generation and permanency across coastal landscapes. A blue carbon indicator (BCI) raster dataset was subsequently generated by combining these rasters together. For the purposes of this study, each of these terms are defined below.

- *Storage*: the volume of blue carbon within coastal Quaternary sediments. Estuaries that are more mature and have expansive alluvial and estuarine floodplains can store larger volumes of fossil blue carbon than coastal barrier sediments that are less favourable for blue carbon storage.
- *Preservation*: the capacity for mangrove and saltmarsh blue carbon to be preserved for long-term sequestration within soils. Saline anaerobic conditions inhibit decomposition. Fine-grained sediments typical of alluvial floodplains, fluvial deltas and, to some extent, estuarine floodplains also inhibit decomposition more than do sandy coastal barrier sediments (Saintilan *et al.* 2013). Fluvial deltas within estuaries are ideal for ongoing preservation of stored carbon because they are composed predominantly of finer grain sizes (although pro-delta and delta fronts may have highly variable grain sizes), and occur in locations influenced by tidal inundation and saline conditions. Consequently, carbon will be more concentrated in these regions. However, coastal barrier sediments are less ideal for carbon storage because they are typically dominated by sands and undergo greater oxidation of sediments (Kelleway *et al.* 2016) and, in some locations, frequent reworking.
- *Generation*: the capacity for existing mangrove forests and saltmarshes to contribute to carbon additionality from living biomass, dead organic material and soil organic carbon. Several studies have indicated that carbon addition is greater in mangrove forests than in saltmarshes (Chmura *et al.* 2003; Pendleton *et al.* 2012), and this is likely to be due to greater height and biomass of mangroves than herbaceous saltmarsh vegetation. In NSW, mangroves forests typically occupy lower positions within the tidal frame than do saltmarshes, and their distribution can be defined on the basis of elevation and hydroperiod (Hughes *et al.* 2019). Additionally, preservation of soil organic carbon within the contemporary range of mangroves has been found to be greater within fine-grained sediments of fluvial origin than within sandy coastal barrier sediments (Kelleway *et al.* 2016).
- *Permanency*: the capacity for carbon to be preserved and not reworked under conditions of higher hydrodynamic energy associated with storms and changes to tidal regimes. The permanency of carbon within substrates has been questioned (DeLaune and White 2012; Kirwan and Mudd 2012), particularly in the context of increased storminess. This component does not specifically indicate retreat pathways for coastal ecosystems as they adapt to sea-level rise. Lower elevations on estuarine shorelines may be exposed to greater hydrodynamic energy because of fetch and wave-action, whereas coastal barrier sediments are more exposed to high wave energy of the open ocean. The exposure of these sediments to higher hydrodynamic energy increases the probability of reworking and poses a risk to carbon permanency.

Human activities in coastal landscapes exert both direct and indirect pressures on blue carbon (Mcleod *et al.* 2011). Rogers *et al.* (2019*b*) accounted for this pressure by using land-use mapping, with the premise being that natural landscapes are more compatible with storage, preservation and generation of blue carbon, whereas intensive land-use activities are less compatible. They proposed that this approach partly accounts for socio-economic factors that influence blue carbon. In this study, land-use mapping was reclassified on the basis of perceived present-day compatibility with blue carbon to generate a blue carbon compatibility (BCC) raster dataset. Combining the BCC and BCI rasters together subsequently provided an indication of blue carbon potential (BCP), as follows:

$$\mathrm{BCI} \times \mathrm{BCC} = \mathrm{BCP}$$

Blue carbon resources: BCI, BCC and BCP

Geological and morphological datasets were used as proxy indicators of blue carbon storage, preservation, generation and permanency. Because this study was undertaken at a regional scale and focused on coastal landscapes rather than individual ecosystems, a trade-off between resolution and spatial extent was essential. Accordingly, the primary input datasets were elevation data derived from the Shuttle Radar Topography Mission and Quaternary and bedrock geology mapping.

- *Shuttle Radar Topography Mission* applied interferometric synthetic aperture radar (InSAR) approach to generate digital elevation models globally. The radar system was deployed in February 2000 and collected data for an 11-day period. Data have been processed and gaps filled by using data derived from the ASTER Global Digital Elevation Model (ASTER GDEM). For Australia, these DEMs derived from SRTM are available at 1° arc-second resolution, equating to a cell size of ~30 × 30 m. For this study, the DEM product, representing ground surface topography with vegetation feature removed, was accessed from Geosciences Australia (https://elevation.fsdf.org.au/). Because this dataset has the lowest resolution of all input datasets, all subsequent datasets were converted to this resolution and cell positions aligned to this dataset. The SRTM-DEM does not reliably indicate elevations below 0 m Australian Height Datum (AHD); consequently, the first-pass assessment focused only on landscape surfaces higher than 0 m AHD. Fortunately, this elevation also approximates the lower limit of mangrove-vegetation distribution.
- *Coastal Quaternary and bedrock geology mapping* (Troedson *et al.* 2004) has been undertaken as part of the NSW Comprehensive Coastal Assessment. This high-resolution mapping classifies depositional units (primarily alluvial plain, estuarine plain and coastal barrier), distinguishes sediment types, processes and geomorphic features (e.g. dune, swamp or channel) and differentiates units by age (i.e. Holocene or Pleistocene). This vector-based dataset can be accessed from online depositories and was reclassified as a raster dataset, with resolution and alignment corresponding to the SRTM-DEM dataset.

Spatial analysis was delimited by the extent of the Quaternary geology dataset (Troedson *et al.* 2004). The coverage of this dataset is restricted to the east by NSW coastline, and to the west by the extent of 1:100 000 map sheets. This somewhat arbitrary western limit results in this dataset not covering all coastal catchments of NSW. This was particularly evident on the northern coast of NSW where large catchments extend farther west than they do on the southern coast of NSW (Fig. 1*c*, *d*). This limitation restricted assessment of blue carbon to the area of catchments within the mapping extent of the Quaternary geology mapping, rather than their full extent. Choropleth raster maps were prepared to indicate blue carbon storage, preservation, generation and permanence. These maps were generated according to geological and morphological criteria and involved reclassifying and adding map layers together according to the cell values in Supplementary Table S1; this was undertaken using the raster-calculator tool on ARCGIS.

To characterise the combined biophysical factors related to blue carbon within coastal landscapes, a blue carbon indicator (BCI) choropleth map was prepared using the raster-calculator tool to add the prior choropleth maps of blue carbon storage, preservation, generation and permanency together. Resulting cell values ranged from 0 to 12. To assist with interpretation of the BCI map and reduce bias from classification, the generated choropleth map was reclassified to produce a final BCI map by using the equal interval classification.

Socio-economic factors, indicated by land-use mapping, may provide additional benefit or risk for blue carbon storage, preservation, and generation. We initially considered the influence of land-use on blue carbon by comparing the area of each BCI class to the area of land-use categories; this aided identification of the land-use classes most compatible with blue carbon. The additional benefits or risks associated with land use was incorporated by converting vector-based land-use maps to raster datasets and on the basis of major land-use categories (accessed at the NSW Government environmental data portal: www.seed.nsw.gov.au), including grazing, conservation area, tree–shrub cover, urban, cropping, river and drainage, wetland, transport–communications, horticulture, mining–quarrying, animal production, and power generation. Using the 2017 land-use map, land-use was reclassified as a raster dataset to a resolution and alignment corresponding to the SRTM-DEM, and cell values were subjectively adjusted on the basis of the perceived compatibility of land use with blue carbon services, as indicated in Table S2. To rationalise blue carbon values and compatibility, BCI and BCC raster datasets were multiplied to provide an overall indication of where opportunities for enhancing or preserving blue carbon services are located. To aid interpretation and reduce bias from the classification, BCP datasets were reclassified using the equal interval classification. Results were interpreted in the context of the total area within a catchment and the proportional area within a catchment for BCI, BCC and BCP.

Influence of barriers on tidal exchange

Wetland drainage and flood mitigation works have had a profound influence on hydrology, especially hydroperiod and tidal exchange. It was rationalised that in-stream barriers below known natural tidal limits impede tidal flows and increase risks of loss of blue carbon services upstream of the barrier. The influence of barriers on tidal exchange was determined by identifying barriers that were located within former tidal limits; this required access to data on barriers and tidal limits.

- *Barriers* or in-stream artificial tidal impediments that may limit blue carbon opportunities were selected from the NSW Government Fish Passage Dataset. This dataset indicates the location of in-stream structures or barriers that obstruct fish passage and, likewise, may impede tidal exchange across NSW. This dataset was provided by the Department of Primary Industries-Fisheries.
- *Tidal limits* were mapped by the NSW Government between 1996 and 2005 to aid management of coastal zones and provide a historical baseline on the location of tidal limits for future monitoring programs (Manly Hydraulics Laboratory 2012). These tidal limits are provided as latitude and longitude and were converted to a point dataset.

Some manipulation of data was necessary because of geospatial errors in the position of some tidal barriers and flow paths. A 1-km buffer was identified at each tidal limit, and barriers within this buffer were considered to serve as a tidal impediment. Expert opinion from NSW Government Department of Primary Industries Fisheries officers verified the position of tidal barriers and their impact as a tidal impediment. A full list of creeks and rivers in which barriers were identified to have a significant influence on tidal exchange is provided in Table S3.

The ARCGIS Hydrology toolset was applied to the SRTM_DEM to model the flow of water across the surface. The 'Fill' tool was used to fill sinks in the SRTM_DEM to remove small depressions that limit the effectiveness of the flow-modelling tools. The 'Flow Direction' tool was used to create a raster dataset representing direction of flow from each cell to its steepest downslope neighbour. The 'Flow Accumulation' tool was used to establish flow paths that were regarded to be rivers, creeks and streams. Some adjustment, again based on expert opinion, was required because drainage works on the low-elevation coastal floodplain have caused the redirection of flow for some waterways. The 'Stream Order' tool was used to identify primary and secondary streams. The positions of tidal barriers that had been adjusted on the basis of expert opinion were subsequently used to establish pour points using the 'Pour Point' tool; hydrological flow from the catchment above this pour point can subsequently be determined. The 'Watershed' tool was subsequently used to delineate watersheds, or drainage area, above the tidal impediments by using the established 'Pour Points'. The watershed above each pour point was named according to the tributary that it is positioned on and was used to extract the area of BCP above each pour point; this indicated the BCP area likely to be influenced by tidal impediments.

Statistical analyses

The area of BCI, BCC and BCP was calculated for each catchment and for each watershed above an in-stream barrier. This conversion provided insight into the tidal impediments that significantly influenced BCP, with the premise being that those with the greatest area could be prioritised for restoration, as they will likely yield greater blue carbon benefits from reinstating tidal exchange. Statistical analyses were initially undertaken to identify whether relationships existed between the generated raster datasets and catchment size by using regression analyses. The premise of these analyses was that catchment size was proportional to blue carbon services. These analyses focussed on the extent of high BCI, high BCC and high BCP because the total area of BCI, BCC and BCP largely corresponds to the extent of Quaternary geology mapping and serves little benefit for decision-making. Full factorial analyses of variance were also used to determine whether geomorphological characteristics of estuaries predicted the observed patterns in high BCI, BCC and BCP. Preliminary results indicated that log-transformation of catchment area and high BCI, BCC and BCP improved statistical models and all analyses were undertaken using log-transformed data. We specifically tested whether a relationship could be established between the area of high BCI, BCC and BCP, and estuary type. Roy *et al.* (2001) classified all estuaries in NSW as the following types: (I) bays, (II) tide-dominated estuaries, (III) wave-dominated estuaries, (IV) intermittent estuaries, and (V) freshwater bodies. Because Type-I estuaries were not included in the study area, there was only one Type-II estuary, and two Type-V estuaries, this analysis focussed on differences arising between estuaries of Type-III and Type-IV. In doing so, this analysis effectively considers the influence of estuary or catchment size on blue carbon. We also tested the relationship between the area of high BCI, BCC and BCP, and estuary maturity. Estuary maturity has also been identified by Roy *et al.* (2001), with each estuary being classified as (A) youthful, (B) intermediate, (C) semi-mature, or (D) mature. Analyses were undertaken on all estuaries within the study area and separated into analyses focussed on the northern coast and southern coast estuaries.

Results

Blue carbon resources: BCI, BCC and BCP

The Clarence, Macleay and Richmond rivers of the NSW northern coast generally had the greatest areas of high storage, preservation, generation, and permanency (Table 1). Although 4 of the 10 largest rivers by catchment are located on the southern coast of NSW (i.e. Shoalhaven, Bega, Tuross and Clyde), it was only the Shoalhaven River that was found to have reasonably high storage (8th highest), generation (6th) and permanency (10th). Consequently, the greatest area with a high BCI occurs predominantly in catchments of the NSW northern coast (Fig. 3*a*). High BCC was greatest in extent in the catchments of the Richmond and Clarence rivers, and catchments with the greatest area of high BCC were also on the northern coast (Table 1). For most catchments, the high category of BCC generally relates to floodplain area and correlates with catchment size. A particular exception is the Richmond River, which has a floodplain area similar in size to that of the Clarence River, yet its catchment area is approximately one-third the size of that of the Clarence River (Fig. 3*b*). The combination of BCI and BCC meant that the most extensive high BCP was largely restricted to estuaries of the NSW northern coast (Fig. 3*c*, Table 1). Detailed quantification of BCI, BCC and BCP is provided in Tables S4 and S5.

The proportions of BCI, BCC and BCP relative to the catchment size are provided in Table S6. Of the top-20 ranked catchments, catchments with the highest proportion of high BCI were generally relatively small (i.e. mostly <5000 ha), although there were some exceptions, including the Tweed River (107 748 ha), Brunswick River (22 993 ha), Tilligerry Creek (135 622 ha) and Cudgen Creek (7076 ha; Fig. S2). Similarly, catchments with the highest proportion of high BCC were also relatively small, except for the Tweed River, Brunswick River, Tilligerry Creek, and Cudgen Creek. Catchments with the highest proportion of high BCP largely had relatively small areas, except for Wooli Wooli River (18 374 ha) and Cathie Creek (11 925 ha). Although the larger systems of the Tweed River, Brunswick River, Tilligerry Creek and Cudgen Creek had high proportions of high BCI and high BCC, the coincidence of these areas was evidently low, and these large catchments did not exhibit remarkably high proportions of high BCP.

The rivers with the largest floodplain areas, that is the Clarence, Macleay and Richmond rivers, overwhelming have the highest areas for storage, preservation, generation and permanency of blue carbon, and this results in a large total BCI area (e.g. Fig. 4 for the Clarence River). The broad coastal floodplains of these rivers, with large spatial extents, are ideal for agriculture and other land-uses, and this is reflected in high total BCI scores; however, there remain large areas within these catchments that have a high BCC area. The outcome of this is that a high BCP area is associated primarily with the larger catchments and particularly those with large floodplains. Only one estuary

Table 1. Catchments with the greatest area with high storage, high preservation, high generation and high permanency scores.

Rank	Catchment (area, ha)	High storage (area, ha)	High preservation (area, ha)	High generation (area, ha)	High permanency (area, ha)	High BCI (area, ha)	High BCC (area, ha)	High BCP (area, ha)
1	Clarence River (2 218 742)	Clarence River (22 980)	Clarence River (8651)	Clarence River (9149)	Richmond River (136 290)	Clarence River (39 445)	Richmond River (43 471)	Clarence River (7877)
2	Hunter River (2 141 399)	Richmond River (21 621)	Richmond River (7498)	Macleay River (6284)	Clarence River (86 753)	Richmond River (29 860)	Clarence River (36 345)	Richmond River (4711)
3	Macleay River (1 131 867)	Hunter River (9874)	Manning River (5555)	Hunter River (5672)	Manning River (53 118)	Hunter River (17 841)	Hastings River (23 635)	Hunter River (3365)
4	Manning River (815 922)	Manning River (7233)	Hastings River (5516)	Richmond River (5443)	Macleay River (49 734)	Macleay River (13 302)	Myall River (19 472)	Macleay River (3361)
5	Shoalhaven River (711 772)	Tweed River (5461)	Hunter River (5293)	Manning River (4879)	Hastings River (42 205)	Manning River (11 829)	Macleay River (19 092)	Hastings River (2291)
6	Richmond River (690 022)	Macleay River (5001)	Macleay River (5219)	Shoalhaven River (3860)	Hunter River (41 478)	Hastings River (8127)	Wallis Lake (16 744)	Shoalhaven River (1388)
7	Hastings River (368 853)	Hastings River (4990)	Wallis Lake (4042)	Hastings River (3368)	Wallis Lake (32 139)	Tweed River (8059)	Hunter River (13 136)	Wallis Lake (1313)
8	Bega River (194 021)	Shoalhaven River (4700)	Tilligerry Creek (3791)	Wallis Lake (2106)	Myall River (25 027)	Shoalhaven River (7806)	Manning River (11 066)	Manning River (845)
9	Tuross River (182 928)	Wallis Lake (2774)	Tweed River (3166)	Tweed River (1796)	Nambucca River (17 058)	Wallis Lake (3594)	Port Stephens (8110)	Port Stephens (596)
10	Clyde River (174 046)	Bellinger River (2044)	Port Stephens (2990)	Myall River (1505)	Shoalhaven River (15 452)	Nambucca River (3049)	Tilligerry Creek (7347)	Myall River (584)

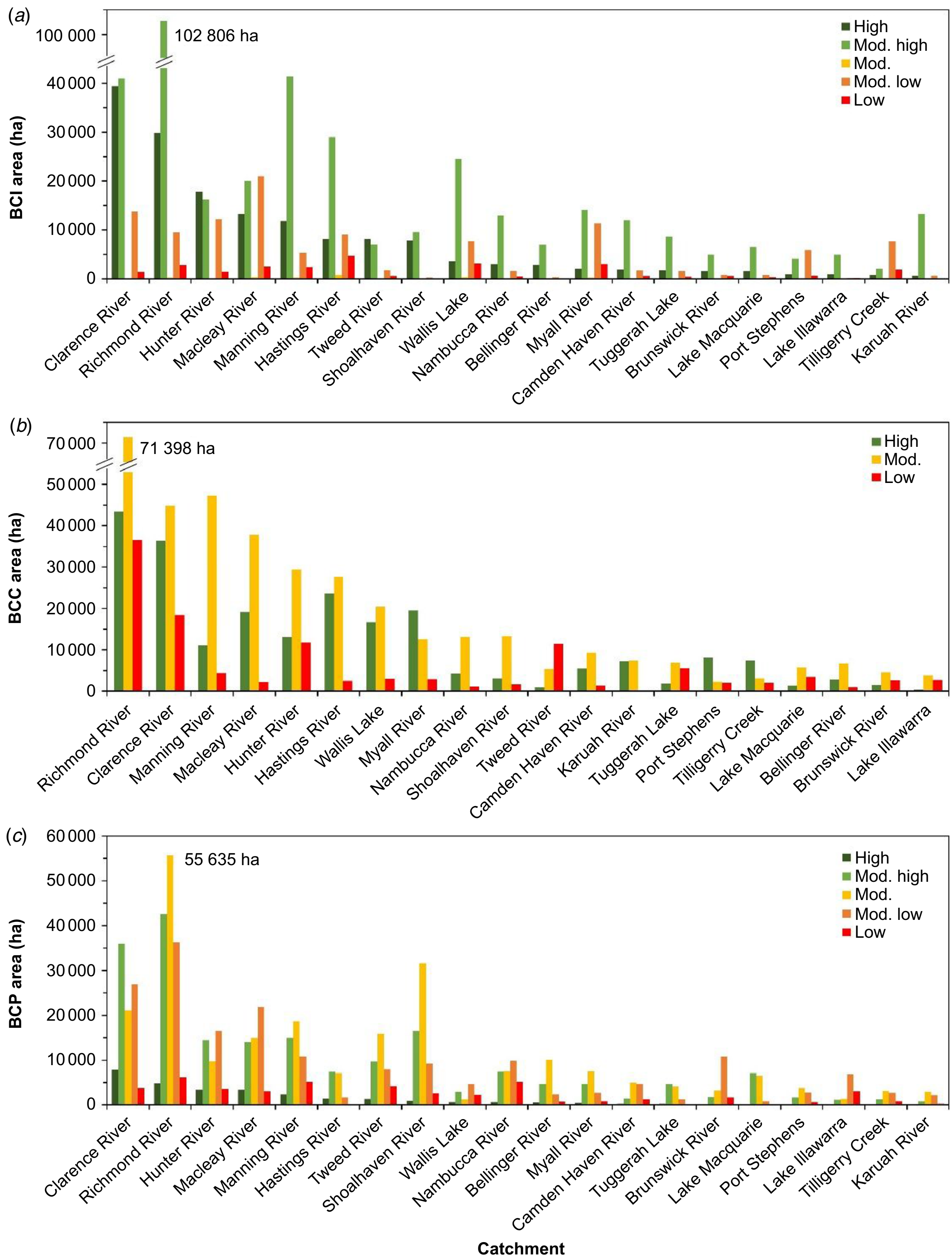

Fig. 3. (*a*) BCI area of low, moderately low, moderate, moderately high and high value within catchments with a large BCI area (i.e. top 20 catchments based on BCI area); (*b*) BCC area of low, moderate, and high value within catchments with a large BCC area (i.e. top 20 catchments based on BCC area); and (*c*) BCP area of low, moderate, and high value within catchments with a large BCP area. Catchments have been ranked from largest to smallest.

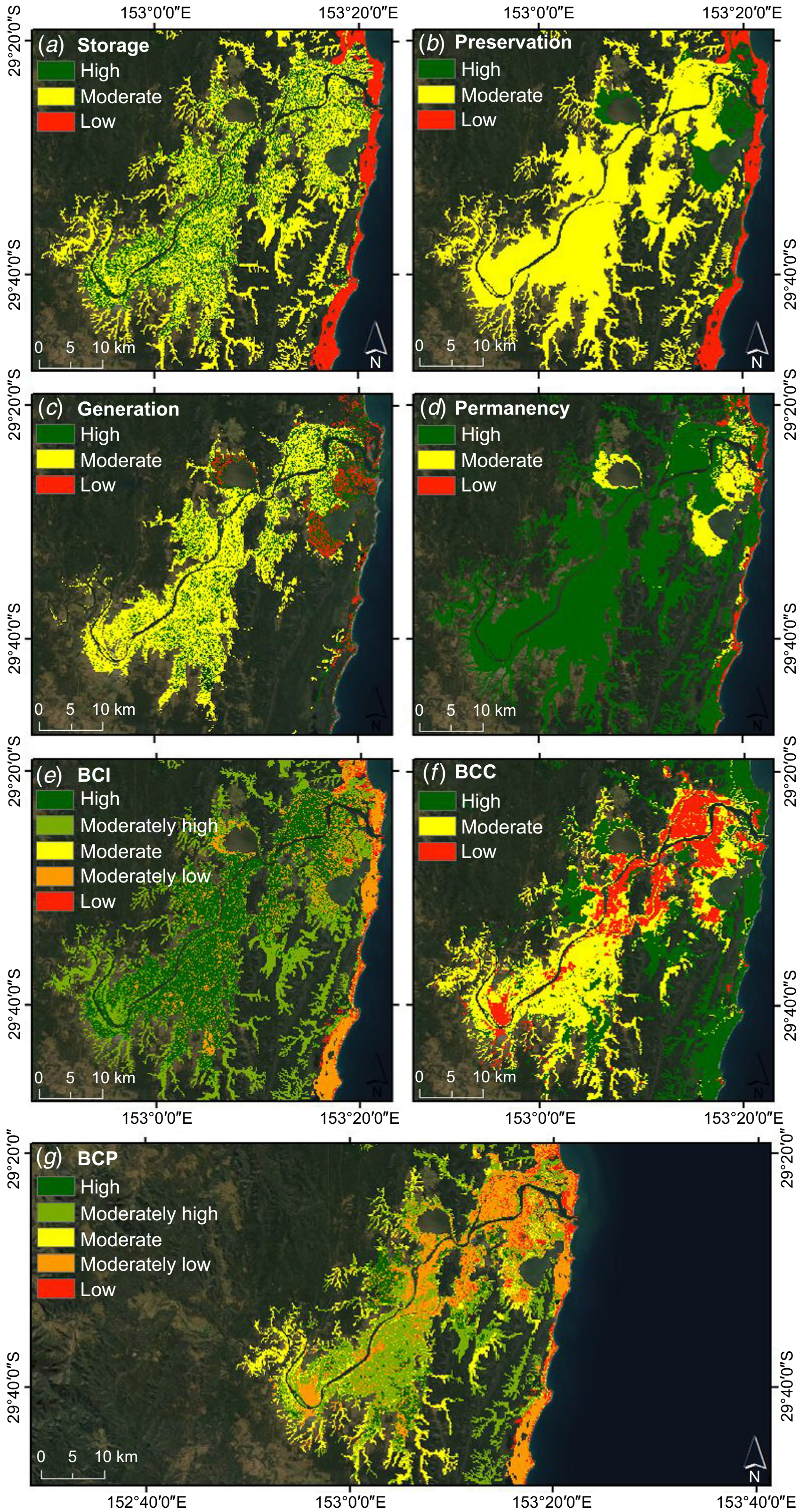

Fig. 4. (*a*) Storage, (*b*) preservation, (*c*) generation, (*d*) permanency of blue carbon, and (*e*) BCI, (*f*) BCC, and (*g*) BCP distribution on the Clarence River.

on the southern coast, the Shoalhaven River, was determined to have a large area of high BCP, and the striking absence of southern coast estuaries relates to the predominance of smaller estuaries, particularly ICOLLs (Table 1).

Comparison of the area of land-use classes in 2017 (Fig. 3a) with an associated BCI indicates an unsurprising relationship among land-use classes of wetlands, and river and drainage systems, and a high BCI area (Fig. 5a). However, a significant area of high BCI coincides with cropping land-use. This indicates that large areas currently used for cropping (mainly sugar cane) are at threat from sea-level rise, but these areas also hold significant opportunities for restoration of blue carbon services and associated co-benefits.

Regression analyses confirmed significant positive relationships between the catchment area and high BCI, BCC and BCP for all estuaries in the study area (Fig. 6a–c, $P < 0.0001$ for all analyses). The greater proportion of large catchments on the northern coast influenced the behaviour of this relationship, and exponential regressions performed significantly better for estuaries of the northern coast (Fig. 6d–f, $P < 0.0001$ for all analyses). The predominance of smaller catchments on the southern coast are likely to have improved the performance of

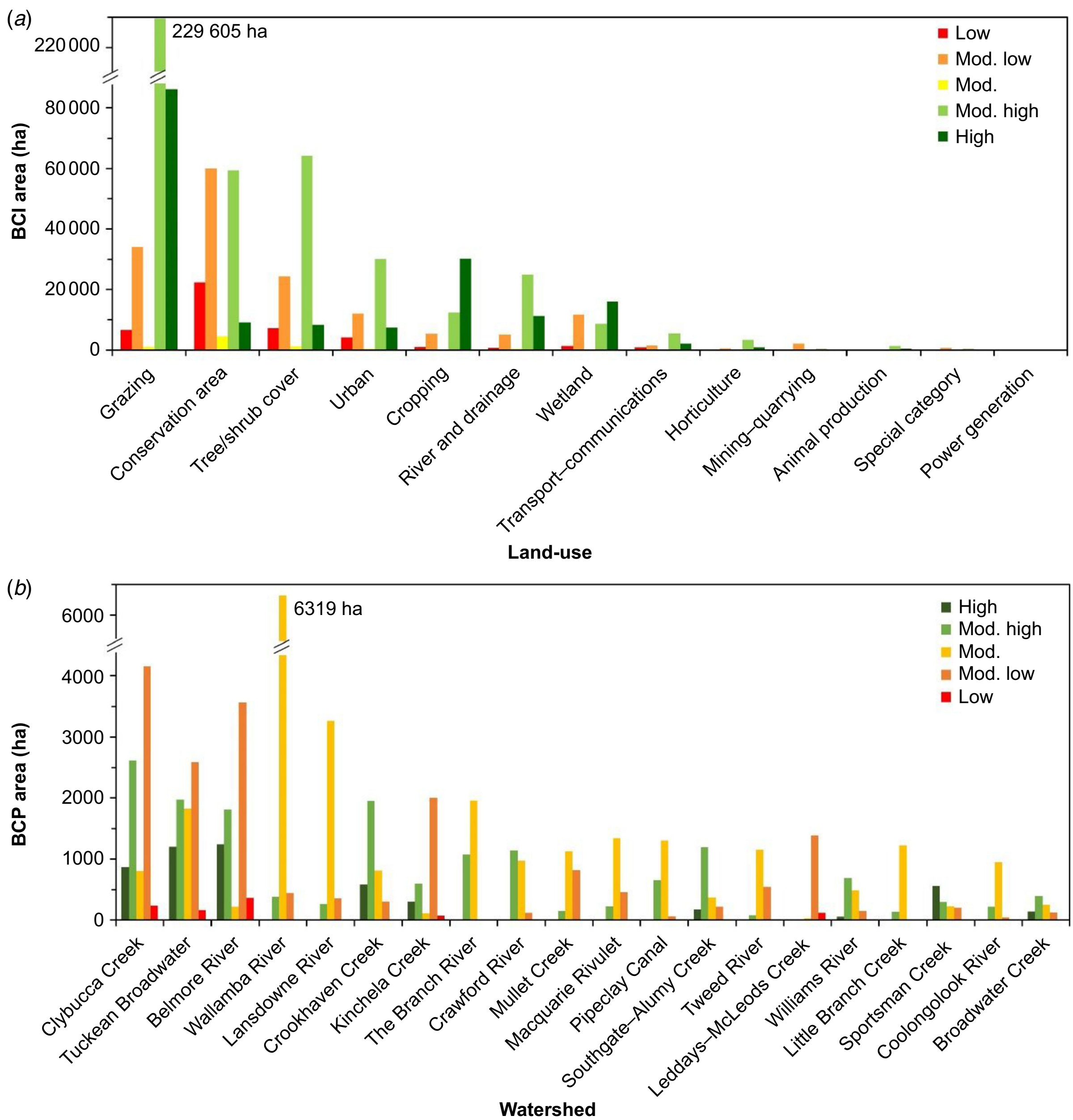

Fig. 5. (a) Coincidence of BCI values and land-use across the study area. The greatest extent of total BCI coincided with grazing and conservation areas, whereas cropping had the greatest extent of high BCI area; and (b) the extent of BCP within watersheds located upstream of a tidal barrier (areas have been ranked from largest to smallest and figures show the top-20 tributaries on the basis of the BCP area within watersheds).

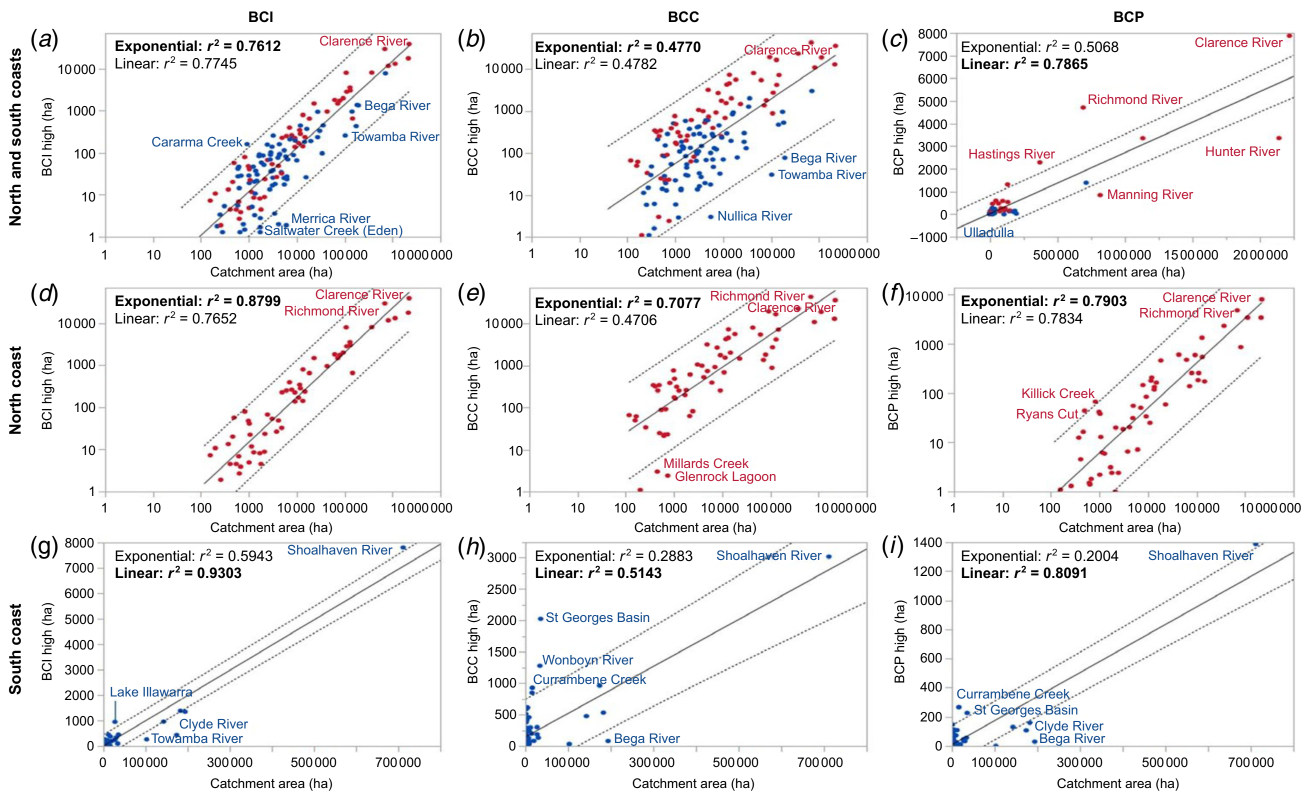

Fig. 6. Relationships between the catchment area on the northern and southern coasts of NSW and (*a*) high BCI, (*b*) high BCC, and (*c*) high BCP; the catchment area on the northern coast and (*d*) high BCI, (*e*) high BCC, and (*f*) high BCP; and the catchment area on the southern coast and (*g*) high BCI, (*h*) high BCC, and (*i*) high BCP. Catchments on the northern coast are indicated by red points, and catchments on the southern coast by blue points. Dashed lines indicate 95% confidence intervals of individuals.

linear regressions (Fig. 6*g*–*i*, $P < 0.0001$ for all analyses). Full factorial analyses that accounted for variation in estuary type and maturity marginally improved on linear regression analyses but did not improve relationships when the large catchments of the northern coast were incorporated. A detailed summary of regression analysis results is provided in Table S7.

Influence of barriers on tidal exchange

Approximately 6074 ha of high BCP area occurs in watersheds above constructed tidal impediments, of which 5154 ha is situated in the northern coast and 920 ha is situated on the southern coast of NSW. The most extensive areas of high BCP in a watershed occur on Belmore River (1240 ha) in the Macleay catchment, Tuckean Broadwater (1199 ha) in the Richmond catchment and Clybucca Creek (866 ha), also in the Macleay. These watersheds also have the most extensive area of all BCP classes (Table 2), and regions where significant gains in blue carbon services and associated co-benefits may be achieved through management of drainage infrastructure and tidal barriers.

Tuckean Broadwater exhibits extensive high and total BCI, and this largely arises from a predominance of moderate to high cell values across storage, preservation, generation and permanency layers (Fig. 7). Coupled with an extensive area of high–moderate to high BCC, Tuckean Broadwater represents an ideal barrier for management, re-engineering or removal to improve blue carbon services. The potential for high blue carbon services should be a factor for management of tidal barriers. It is especially significant at Clybucca Creek, Belmore River, Wallamba River in the Wallis Lake catchment, Sportsmans Creek in the Clarence catchment and Crookhaven Creek in the Shoalhaven catchment. For detailed quantification of the area of blue carbon storage, preservation, generation, permanency, BCI, BCC and BCP by watershed, see Tables S8 and S9.

Discussion

Geomorphology as a control on blue carbon

There is increasing awareness of intertidal position (Cacho *et al.* 2021), sediment character (Kelleway *et al.* 2016; Gorham *et al.* 2021) and local-scale geomorphology (van Ardenne *et al.* 2018) as controls on mangrove and saltmarsh blue carbon storage. Local-scale analyses have

Table 2. Tributaries with the largest area above a barrier of high BCI, total BCI, high BCC, total BCC, high BCP and total BCP.

Rank	High BCI (area, ha)	Total BCI (area, ha)	High BCC (area, ha)	Total BCC (area, ha)	High BCP (area, ha)	Total BCP (area, ha)
1	Tuckean Broadwater, Bagotville barrage, Tuckean Wetland, **Richmond** (3253)	Clybucca Creek Menarcobrinni floodgate, Seven Oaks Drain, **Macleay** (8680)	Belmore River, Belmore Swamp, **Macleay** (4870)	Clybucca Creek, Menarcobrinni floodgate, Seven Oaks Drain**,** **Macleay** (9578)	Belmore River, Belmore Swamp, **Macleay** (1240)	Clybucca Creek, Menarcobrinni floodgate, Seven Oaks Drain, **Macleay** (8680)
2	Crookhaven Creek, Culburra Road floodgate, **Shoalhaven** (2429)	Tuckean Broadwater, Bagotville barrage, Tuckean Wetland, **Richmond** (7747)	Clybucca Creek, Menarcobrinni floodgate, Seven Oaks Drain, **Macleay** (3744)	Tuckean Broadwater Broadwater, Bagotville barrage, Tuckean Wetland, **Richmond** (8155)	Tuckean Broadwater, Bagotville barrage, Tuckean Wetland, **Richmond** (1199)	Tuckean Broadwater, Bagotville barrage, Tuckean Wetland, **Richmond** (7747)
3	Clybucca Creek Menarcobrinni floodgate, Seven Oaks Drain, **Macleay** (2415)	Belmore River, Belmore Swamp, **Macleay** (7205)	Tuckean Broadwater, Bagotville barrage, Tuckean Wetland**,** **Richmond** (2528)	Belmore River, Belmore Swamp, **Macleay** (7808)	Clybucca Creek, Menarcobrinni floodgate, Seven Oaks Drain, **Macleay** (866)	Belmore River, Belmore Swamp, **Macleay** (7205)
4	Belmore River, Belmore Swamp, **Macleay** (2195)	Wallamba River, Clarksons crossing, **Wallis** (7141)	Kinchela Creek, Swan Pool, **Macleay** (1717)	Wallamba River, Clarksons crossing, **Wallis** (7136)	Crookhaven Creek, Culburra Road floodgate, **Shoalhaven** (587)	Wallamba River, Clarksons crossing, **Wallis** (7136)
5	Southgate–Alumy Creek, **Clarence** (1400)	Lansdowne River, Lansdowne Weir, **Manning** (3884)	Crawford River Bulahdelah, **Myall** (1120)	Lansdowne River, Lansdowne Weir, **Manning** (3883)	Sportsman Creek, Sportsmans Creek Weir, Everlasting Swamp, **Clarence** (556)	Lansdowne River, Lansdowne Weir, **Manning** (3883)
6	Leddays–McLeods Creek, **Tweed** (967)	Crookhaven Creek, Culburra Road floodgate, **Shoalhaven** (3666)	The Branch River, **Karuah** (1074)	Crookhaven Creek Culburra Road floodgate, **Shoalhaven** (3688)	Coldstream River**,** **Clarence** (343)	Crookhaven Creek, Culburra Road floodgate, **Shoalhaven** (3657)
7	Kinchela Creek, Swan Pool, **Macleay** (808)	Kinchela Creek, Swan Pool, **Macleay** (3086)	Crookhaven Creek, Culburra Road floodgate, **Shoalhaven** (896)	Kinchela Creek, Swan Pool, **Macleay** (3131)	Kinchela Creek, Swan Pool, **Macleay** (308)	Kinchela Creek, Swan Pool, **Macleay** (3086)
8	Sportsman Creek Sportsmans Creek Weir, Everlasting Swamp, **Clarence** (740)	The Branch River, **Karuah** (3045)	Sportsman Creek, Sportsmans Creek Weir, Everlasting Swamp, **Clarence** (846)	The Branch River, **Karuah** (3045)	Crookhaven River, Culburra Road floodgate, **Shoalhaven** (199)	The Branch River, **Karuah** (3045)
9	Williams River, Seahams Weir, **Hunter** (658)	Crawford River, Bulahdelah, **Myall** (2246)	Pipeclay Canal, Big Swamp, **Manning** (609)	Mullet Creek, **Lake Illawarra** (2269)	Southgate–Alumy Creek, **Clarence**(169)	Crawford River, Bulahdelah, **Myall** (2245)
10	Coldstream River, **Clarence** (484)	Mullet Creek, **Lake Illawarra** (2113)	Broadwater Creek, The Broadwater, **Clarence** (527)	Crawford River, Bulahdelah, **Myall** (2263)	Poverty Creek, **Clarence** (157)	Mullet Creek, **Lake Illawarra** (2105)

Catchments are indicated in bold.

become the foundation on which spatial frameworks for projecting blue carbon services have been developed (Rogers *et al.* 2019*b*). Additionally, landscape-scale (Ewers Lewis *et al.* 2020), national (Cameron *et al.* 2021) and global-scale analyses (Rovai *et al.* 2018; Twilley *et al.* 2018) have confirmed that mangrove and saltmarsh blue carboon is strongly related to estuarine geomorphology. Using the framework of Rogers *et al.* (2019*b*), this study demonstrated that blue carbon services, indicated by BCI, are proportional to the catchment area on both the northern coast and southern coast of NSW, and are controlled by geomorphology. Relationships between BCI and catchment area were improved when accounting for estuary type and the maturity. Moreover, we found that the relationship between BCI and catchment area was best predicted on the basis of an exponential model on the northern coast and a linear model on the southern coast of NSW. This difference in model structure largely arises

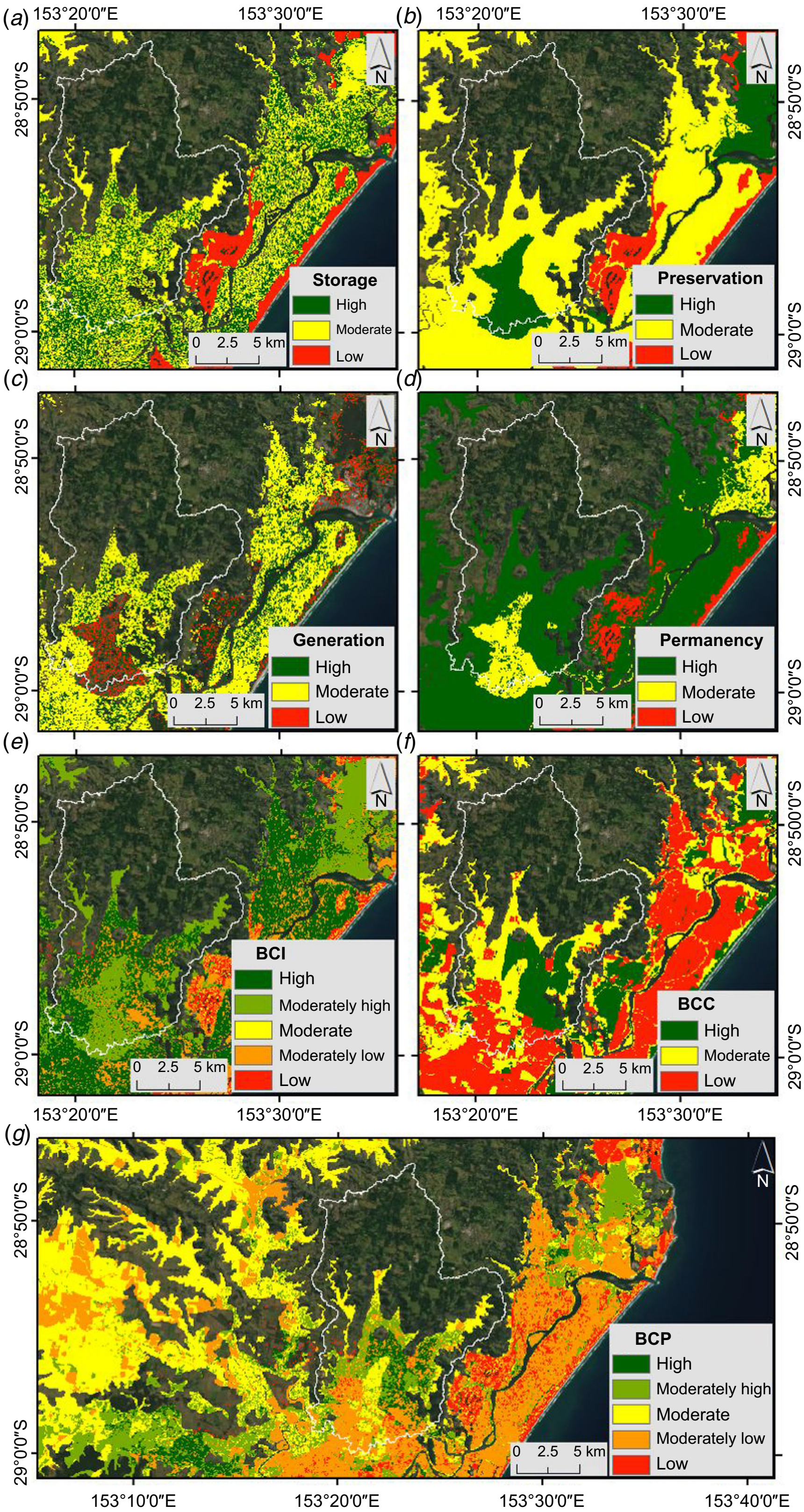

Fig. 7. (*a*) Storage, (*b*) preservation, (*c*) generation, (*d*) permanency, and (*e*) BCI, (*f*) BCC and (*g*) BCP within Tuckean Broadwater (indicated by white boundary). This represents a substantial area of high BCI and BCP located above a tidal barrier.

from the variation in scale of the coastal floodplains and catchments on the NSW northern and southern coasts. This variation is due to the western boundary of the coastal floodplain along eastern Australia being demarcated by the slopes of The Great Dividing Range, which are positioned further from the coastline on the NSW northern coast than the southern coast. Additionally, the low gradient of the continental shelf north of Newcastle provides ample lateral space for shoreline progradation and the development of broad coastal floodplains (Roy *et al.* 1980, 2001). In contrast, the southern coast floodplain is narrow, and catchments are more numerous and smaller. Large estuaries within NSW provide sufficient storage and preservation of fossil blue carbon and support the addition of contemporary blue carbon from mangrove forests and saltmarshes. The enhanced ability for contemporary blue carbon addition along the NSW northern coast is confirmed by state-wide mangrove and saltmarsh mapping, which indicates that the northern coast supports 8500 ha of mangrove and 4910 ha of saltmarsh habitats, whereas the southern coast supports 1624 ha of mangrove and 1260 ha of saltmarsh (extracted from NSW Estuarine Macrophytes mapping). This study also confirmed the finding of Rogers *et al.* (2019*b*) that high proportion of high BCP was associated with smaller catchments that tend to have more extensive low-lying areas, and a high proportion of the catchment land-use designated as conservation area or wetland.

Extensive coastal floodplains are also ideal for intensive cropping and grazing. The conversion of blue carbon habitats and associated carbon storage to agricultural lands has been facilitated by wetland drainage, tidal impediments and flood mitigation controls. These activities are markedly more extensive on the northern coast of NSW where the coastal floodplain is broad and engineered structures separate larger areas from tidal exchange. Consequently, the compatibility of land-use with blue carbon is also proportional to catchment area. In many instances, land-use activities that are incompatible with blue carbon can occur only because of the presence of engineered structures that serve as tidal barriers. The effect of land-use and land-cover change on blue carbon is well established, with reports of substantial declines in biomass and soil carbon stocks following land-use conversion (Sasmito *et al.* 2019). Management decisions that facilitate restoration of blue carbon ecosystems, achieved by (1) removal or management of tidal barriers, and (2) land-use change to activities compatible with blue carbon will achieve optimal blue carbon outcomes.

Influence of barriers and opportunities for improving blue carbon services

Rehabilitation efforts (i.e. restoration, regeneration and afforestation) can effectively improve biomass carbon stocks and re-establish soil carbon stocks (Sasmito *et al.* 2019). It is for this reason that the reintroduction of tidal flow to facilitate establishment of mangroves, tidal marshes and supratidal forest is regarded as a priority activity that could be undertaken to generate Australian Carbon Credit Units within the ERF (Kelleway *et al.* 2017). This study focussed on identifying land located in watersheds upstream of tidal barriers that could be regarded as priority areas for tidal reintroduction. Because of the occurrence of broad coastal floodplains and associated greater prevalence of tidal barriers on the northern coast, opportunities for reintroduction of tides above tidal barriers are particularly prevalent on the northern coast. More specifically, ~5153 ha with a high BCP are located upstream of tidal barriers on the northern coast and 919 ha are located on the southern coast.

Tidal reintroduction and reinstatement of higher water tables has already commenced in some watersheds of the NSW northern coast and notable examples include partial restorations at Hexham and Tomago swamps on the Hunter River, Big Swamp on the Manning River, and Yarrahapini Wetland on the Macleay River (Rogers *et al.* 2016*b*). The benefits of these activities not only include blue carbon services but also associated co-benefits such as reconnection of fish passage, improvement in water quality associated with the reinstatement of higher water tables and tidal exchange to acid sulfate soils, inhibiting the activation of potential acid sulfate soils and reducing the frequency and intensity of black water events. On the basis of the extent of high BCP area within watersheds above tidal barriers, blue carbon is likely to be a particularly significant factor for barrier management decision-making at Tuckean Broadwater (1199 ha), Clybucca Creek (866 ha), Crookhaven Creek (587 ha) and Sportsmans Creek (556 ha).

Increasing coastal wetland reserves may precede reintroduction of tides and, in many cases, tidal reintroduction is likely to be simpler on publicly owned land (Bell-James and Lovelock 2019*b*). However, achieving optimal blue carbon services and co-benefits not only requires consideration of tenure including Native Title, and land-use, but may also require interventions to ensure restoration of hydrological regimes affected by drainage, delivery of ecosystem services and adaptations to sea-level rise (Abbott *et al.* 2020; Karim *et al.* 2021; Sadat-Noori *et al.* 2021). This is because past agricultural land-use can remain imprinted on coastal wetlands for decades or centuries, with evidence of past drainage, structures and fencing retained in, and sometimes continuing to degrade, post-agricultural landscapes (Williams and Watford 1997; MacDonald *et al.* 2010). These impacts are amplified by changes to biodiversity and soil biogeochemistry that can arise from decades of grazing, cropping and drainage, such as acid sulfate soils impacts. including autocompaction of ground-surface elevations (Johnston *et al.* 2003, 2016) or destruction of peat because of fire. The imprint of past agricultural activities such as drainage ditches and even reasonably minor features such as mosquito runnels can become a conduit for increased tidal flows and supply of mangrove propagules into saltmarsh, altering hydroperiod and the ecological character of wetlands for decades (Breitfuss *et al.* 2003, Knight *et al.* 2021). In some instances, where consideration of the potential impacts of tidal inundation on adjacent land-use is essential, or where acid sulfate soils have been activated, interventions to restore hydrological regimes may be substantial. Interventions can include filling or blocking drains to raise ground-water levels, removal of levees or implementation of 'smart' flood gates to manage tidal regimes, as occurred in rehabilitation projects across the Hunter River (Sadat-Noori *et al.* 2021). To achieve the best delivery of blue carbon services and co-benefits, consideration should be given to restoration of natural hydrological regimes, this may involve bringing forward land-use change that will inevitably occur with sea-level rise.

Policy and legislation in Australia (Rogers *et al.* 2016*a*; Rog and Cook 2017), and particularly NSW, now affords considerable protection to coastal wetlands from any changes to drainage or extent, and have been effective in halting the trajectory of decline in wetland extent that occurred until the 1980s. However, this policy and legislation may not be effective at halting the incremental conversion of adjacent freshwater and brackish wetlands that may be hotspots of fossil blue carbon, or the suppression of tidal flows and sea-level rise with minor, but strategic, filling work. This is evident from the ongoing generation of acid sulfate soils, activated when pyrite oxidises following drainage of carbon stores (Rosicky *et al.* 2004). Additionally, it is not yet effective in protecting fossil blue carbon that is preserved in substrates that may no longer support contemporary mangrove forests and saltmarshes, or adjacent landward zones that may become important retreat pathways as coastal wetlands respond to sea-level rise. In the absence of robust legislation that accommodates and protects coastal wetland retreat pathways as they respond to sea-level rise (Rogers *et al.* 2016*a*), this pattern of land-use conversion may continue.

Prioritising land for blue carbon restoration

The January 2022 approval of a blue carbon reinstatement of tidal flows methodology within the ERF has focussed attention on prioritising land for tidal reintroduction, particularly because this activity is likely to achieve a rapid increase in blue carbon services. The method demonstrated in this study can be used to establish priority areas upstream of in-stream barriers, and the use of data-sets initially developed to prioritise restoration of fish passage may be applied in other jurisdictions to identify projects within the ERF. In particular, high BCP was identified upstream of many in-stream barriers (see Table 2), and these locations will serve as priority areas for further investigation of the feasibility of tidal reintroduction by using the ERF methodology. The blue carbon prioritisation method described here focussed on in-stream barriers and does not explicitly consider opportunities that may arise from managing levees to reintroduce over-land tidal exchange. Use of flood mitigation levee data sets from local councils and use of LiDAR data sets to identify historical and privately built levees and other floodplain structures that impede tidal flows would enable further analyses of hydrological modification caused by levees. These analyses may highlight additional priority areas for further investigation. For example, tidal reintroduction beyond levees has already commenced at Hunter National Park to facilitate creation of shorebird and waterbird habitat (Glamore *et al.* 2021), and the feasibility of an additional ERF project expanding beyond this initial work could also be investigated.

Bell-James and Lovelock (2019*b*) emphasised that difficulties can arise when managing barriers for tidal reintroduction, particularly when tenure is complex and public–private ownership arrangements are required. This is especially the case when ownership and legislative processes operating in the intertidal zone are complex (Rog and Cook 2017; Bell-James and Lovelock 2019*a*); hence, implementation of tidal restoration projects are likely to be expedited when the land targeted for restoration is wholly within public ownership and managed either by local, state or federal government. However, land tenure should not preclude tidal restoration beause it is anticipated that the ERF will incentivise land managers to consider tidal reintroduction for blue carbon services (Kelleway *et al.* 2017, 2020; Macreadie *et al.* 2017*a*), and necessary approvals could be sought to facilitate restoration (Bell-James and Lovelock 2019*b*).

Land upstream of an in-stream barrier that could provide high blue carbon services and co-benefits are regarded to be priority areas for tidal reintroduction and generation of ACCUs within the ERF. This may provide sufficient incentive to facilitate the conversion of very marginal agricultural land to coastal wetland through the reintroduction of tides, particularly where approval processes are streamlined, and success is facilitated (Bell-James and Lovelock 2019*b*) and the alternatives involving maintaining or reconstructing barriers are expensive and jurisdictionally complex.

Tidal barriers, because of their function, have areas upstream within the range of the intertidal zone that could begin developing blue carbon ecosystems following removal of the structure. Agricultural production reliant on tidal barriers will become increasingly marginal and constrained with sea-level rise, and this may make coastal wetland rehabilitation increasingly favourable. Noteworthy is the considerable evidence of declining ecosystem services and increases in ecosystem disservices with ponded pastures (Bell-James and Lovelock 2019*a*) and mosquito hazard (Knight *et al.* 2017), and potential reduction in agricultural productivity with sea-level rise (Park *et al.* 2008; Howden and Crimp 2011). Although this analysis did not explicitly consider land-use planning decisions that would facilitate coastal wetland retreat with sea-level rise, many tidal barriers have not been designed to meet anticipated increases in tidal planes associated with sea-level rise. It is probable that existing tidal barriers may be overtopped, and tidal barriers will no longer hold back the tide, but inhibit drainage of tidal and flood waters, resulting in the development of ponded pastures and associated disservices. Less obvious, but more effective, is how increases in tidal planes reduce drainage opportunities from floodgate structures that were engineered to operate due to the differential head pressure achieved at low tides on the basis of the position of the tidal plane when the structures were first built in the 1950s–1970s. Analyses that incorporate projections of tidal planes with sea-level rise will improve capacity to identify priority areas beyond tidal barriers that will have less ability to drain and a higher risk of becoming increasingly inundated. Here we estimated that low-lying land beyond barriers with elevations of <2 m AHD will become progressively less viable for cropping and agriculture (based on RCP 8.5 sea-level rise projection of ~1- and 2-m tide range, centred at ~0 m AHD). Opportunities for generating ACCUs within the ERF are substantial in these circumstances as both reintroduction of tides, and land-use planning for sea-level rise, through the establishment of retreat pathways, are regarded to be suitable activities within the ERF framework (Kelleway *et al.* 2017) although to date only reinstatement of tidal flows has been developed as a verified methodology in Australia. Reinstating tidal flows and accommodating sea-level rise will also deliver on a suite of co-benefits additional to anticipated carbon abatement that will improve longer-term utility of coastal landscapes, such as inhibiting methanogenic processes (Poffenbarger *et al.* 2011), improving trophic food web provision, fish passage and habitat (Rogers *et al.* 2016*b*), coastal and shoreline protection, nutrient cycling, reduction in blackwater events and improved water quality (Duarte *et al.* 2013).

In some cases, particularly when in-stream barriers were constructed decades ago, tidal barriers may either be failing due to deterioration or may no longer meet design objectives to hold back the tide or drain coastal landscapes to the degree landholders expect and have used to design their agricultural systems. Additionally, ongoing soil diagenesis, organic matter decomposition and soil shrinkage associated with drainage of coastal landscapes can lead to significant loss of substrate elevations (Rosicky *et al.* 2004), sometimes to the extent where saline intrusion through substrates now reaches at or near the surface resulting in acid sulfate soil scald formation (Rosicky *et al.* 2004). The ingress of saline waters beyond tidal barriers is already evident in many locations by the occurrence of saltmarsh in depressions and along abandoned palaeochannels (pers. obs.). In some instances, because ground surface has lowered and sea level has increased, in-stream barriers may not be able to deliver on their initial designed primary purpose of flood mitigation, instead they behave counter to this objective, at times, slowing drainage of freshwater from catchments and leading to ponded pastures, or trapping floodwaters and amplifying flooding impacts. The disservices associated with ponded pastures are well known, including increased emissions of methane (Kroeger *et al.* 2017) and other greenhouse gases (Dalal *et al.* 2008). With the radiative forcing of methane in the atmosphere being 25–100 times greater than carbon dioxide, the creation of ponded pastures is contrary to national efforts to mitigate climate change (Kroeger *et al.* 2017).

When the efficacy of tidal barriers is becoming limited, land managers are left with options to (1) seek approval to retro-fit or re-engineer barriers, an expensive option with risks that inundation impacts will continue to increase with sea-level rise and further ingress of saline water into drained landscapes via sand seams, macropores and palaeochannels; (2) do nothing and accept that tides will increasingly constrain previously achieved rates of drainage and periodically over-top barriers causing agricultural land-use to become less viable because of the trapping of flood and tidal waters; or (3) seek approval to collaborate with structure owners remove tidal barriers, reintroduce tides and restore a natural hydrological regime and secure blue carbon values. Although many existing barriers were established prior to the need for approvals (Creighton *et al.* 2015), changes to existing barriers or the construction of new 'improved' barriers now require approvals. This approval process, and the cost of works, may provide the opportunity for asset owners, land managers and the broader community – when the structure is owned by a public authority – to assess the services and disservices associated with tidal barriers. In some situations, ERF opportunities may become increasingly appealing. Importantly, although a delayed decision, or the 'do nothing' option, will not prevent the inevitable failure of barriers to hold back the tide, it may limit access to ERF opportunities. Registration of an 'activity' to deliberately reinstate tidal flows must be undertaken prior to tidal restoration occurring due to failure of a barrier. Prompt decisions to remove barriers or manage them for reintroduction of tides will increasingly become the most prudent option for asset owners and land managers.

Recommendations for blue carbon opportunities in NSW

Because of the aging of existing structures that serve as tidal barriers and the effects of accelerating sea-level rise on existing land-use, land managers will increasingly be required to make decisions about both existing drainage and tidal barriers and the engineering of new drainage and tidal barriers. Despite the emerging risks that increasing tidal inundation place on existing land-use, considerable opportunities are available to shift the perspective of land productivity and contribute to climate change mitigation; however, this will require a paradigm shift in coastal floodplain management from one that promotes 'holding the tide back' to one that facilitates tidal inundation. The incentives associated with facilitating tidal inundation require timely decisions to ensure that activities that facilitate reintroduction of tides or adaptation of coastal wetlands to sea-level rise are registered within the ERF and implemented in advance of tidal reintroduction or adaptation to sea-level rise. Critically, if an aging in-stream barrier fails prior to project registration and tidal restoration commences prior to implementation of activities, the opportunity for the land manager, or broader community to benefit financially under the ERF, may not be achieved. Consequently, opportunities provided by the ERF may motivate stakeholders to reinstate tidal exchange sooner and, therefore, improve the capacity of land to adapt to sea-level rise prior to significant acceleration in sea-level rise. To fully realise these opportunities and ensure that jurisdictions in Australia are well placed to make timely decisions, we recommend the following:

1. *Auditing the location and condition of all tidal barriers*. With more than 4200 in-stream structures impeding tidal flows in coastal rivers and streams of NSW, there are tremendous opportunities for managing barriers differently. To prioritise opportunities requires more information about the precise location, ownership, land tenure, structure, condition and height of tidal barriers. An audit of tidal barriers, focussing initially on the most significant barriers, will provide decision makers with the essential information to prioritise opportunities and approve activities for reintroduction of tides. Such audits will identify aging structures that no longer meet design objectives, and for which a decision regarding re-engineering or removal should be made in a timely manner and with consideration given to the provision of blue carbon services and other co-benefits.
2. *Quantifying the projected effects of sea-level rise on tidal planes*. Prioritising land above or below tidal barriers that will have tidal reintroduction imposed by sea-level rise requires consideration of future coastal wetland retreat pathways. A range of techniques can be used to identify future retreat pathways. These include reasonably simple indicator techniques (Rogers and Woodroffe 2016) and simple bath-tub modelling. Alternatively, more sophisticated approaches could be used, such as projections of tidal planes (Hanslow *et al*. 2018), geomorphological modelling (Rogers *et al*. 2014; Mogensen and Rogers 2018), or hydrodynamic modelling (Rodríguez *et al*. 2017; Kumbier *et al*. 2018; Khojasteh *et al*. 2021). Integration with tidal plane analyses (Wen and Hughes 2022) may increase confidence in possible future retreat pathways and can be used to identify retreat pathways where decisions can be made now to improve blue carbon futures.
3. *Assessing the efficacy of existing barriers and their drainage units under different sea-level rise scenarios*. Because many structures that impede tides are engineered on the basis of past environmental conditions, it is anticipated that their ability to drain landscapes at the designed rate will diminish. This is because increased sea levels will result in fewer occasions when low tides are below the invert of the floodgate valve, a position when the floodgate outlet drainage function is unimpeded by estuary water levels. Often described as 'losing the low tides', it is a fundamental constraint on drainage from barriers. It is also anticipated that as sea-level rise accelerates, higher tides will intrude through sand seams, macropores and palaeochannels into landscapes previously protected by drainage infrastructure. Concurrently, the efficacy of floodgates and levees in holding back the highest tide will be exhausted once tidal planes over-top the in-stream structures. Integrating information about blue carbon potential with projections of the effects of sea-level rise on tidal planes, and information regarding the condition and dimension of in-stream structures, will provide the capacity to identify structures where tides will constrain operation or over-top under sea-level rise scenarios, and when this effect is likely to occur. This information will be crucial for prioritising management of barriers in advance of sea-level rise.
4. *Developing decision-support tools for evaluating economic and environmental costs and benefits of tidal-barrier decisions*. Although the ERF provides the mechanism to apply an economic value to environmental benefits provided by coastal wetlands, it does not provide the basis for adequately incorporating this into decisions regarding the ongoing management of tidal barriers, particularly where a change in land-use is incurred. Assessments of agricultural financial impact coupled with the public environmental benefit associated with land-use change are rare but do exist (see, for example, Beardmore *et al*. 2019). Conservation planning tools have been applied in the context of restoration for ecosystem services (Adame *et al*. 2015; Carwardine *et al*. 2015; Gilby *et al*. 2021). These tools could be modified to facilitate the adequate and fair assessment of costs and benefits associated with change in land-use, design and construction of barriers, and provision of blue carbon services and other co-benefits.
5. *Establishing policy for approving upgrades of existing or construction of new tidal barriers that accounts for blue carbon and other co-benefits*. An increase in requests to re-engineer existing structures or construct new structures is likely because structures age and anticipated sea-level rise accelerates. Equipping decision-makers with a decision-making framework will ensure that opportunities to mitigate climate change are realised. In NSW, the *State Environmental Planning Policy (Resilience and Hazard)* 2021 (see https://legislation.nsw.gov.au/view/html/inforce/current/epi-2021-0730) (RH SEPP) provides a useful foundation. Decisions are also, in part, informed by the document *Policy and guidelines for fish habitat conservation and management (update 2013)* (see https://www.dpi.nsw.gov.au/fishing/habitat/

publications/pubs/fish-habitat-conservation; Fairfull 2013). This document largely focuses on maintaining fish passage by the design and construction of in-stream structures and the rehabilitation of barriers to fish passage. Although effective in meeting these objectives, it does not provide guidance that will facilitate decisions that improve or restore blue carbon services. The absence of a framework that integrates fish passage, blue carbon services and other co-benefits (i.e. ecosystem services) may hamper opportunities for achieving payment for ecosystem services from the ERF sufficient to motivate some current landholders. Furthermore, such decision-making needs to consider impacts on vegetation communities, including threatened ecological communities that may have developed in response to hydrological changes caused by installation of the barrier, and which may be subsequently inundated when reintroduction of tides occurs, cognisant of the likely inevitable loss of these communities with sea-level rise.

This study has prioritised watersheds above tidal barriers that are ideal locations for carbon offsetting within the ERF by using the blue carbon tidal-restoration methodology. Significant opportunities on the coastal floodplains of northern NSW are available for managing tidal barriers differently, restoring tidal flows and rehabilitating floodplains to natural habitats vegetated with blue carbon ecosystems (mangrove, saltmarsh and supratidal forest). The Commonwealth of Australia is seeking to increase carbon stocks and improve reporting to UNFCCC, and the development of a reinstatement of tidal flow methodology to support payment for blue carbon restoration and management (Kelleway *et al.* 2020) will further incentivise the conversion of degraded coastal habitats to high-priority blue carbon areas. Critically, accessing incentives associated with facilitating tidal inundation requires collaboration and timely decision-making to ensure that activities that facilitate reintroduction of tides or adaptation of coastal wetlands to sea-level rise are registered within the ERF and implemented in advance of tidal reintroduction or adaptation to sea-level rise.

Supplementary material

Supplementary material is available online.

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Data availability. Spatial data generated from this analysis are available on the NSW SEED website; the Central Resource for Sharing and Enabling Environmental Data in NSW, or from the authors by request.

Conflicts of interest. Kerrylee Rogers is an editor for *Marine and Freshwater Research* but did not at any stage have editor-level access to this manuscript while in peer review, as is the standard practice when handling manuscripts submitted by an editor to this journal. *Marine and Freshwater Research* encourages its editors to publish in the journal and they are kept totally separate from the decision-making processes for their manuscripts. The authors have no further conflicts of interest to declare.

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Author affiliations

[A]School of Earth, Atmospheric and Life Sciences, University of Wollongong, Wollongong, NSW 2522, Australia.

[B]Science, Economics and Insights Division, NSW Department of Planning and Environment, Lidcombe, NSW 2141, Australia.

[C]Coastal Systems, NSW Department of Primary Industries Fisheries, Wollongbar, NSW 2477, Australia.